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Scallop dredging has profound, long-term impacts on maerl habitats

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Maerl beds are mixed sediments built by a surface layer of slow-growing, unattached coralline algae that are of international conservation significance because they create areas of high biodiversity. They are patchily distributed throughout Europe (to ~30 m depth around the British Isles and to ~120 m depth in the Mediterranean) and many are affected by towed demersal fishing. We report the effects of Newhaven scallop dredges on a previously unfished maerl bed compared with the effects on similar grounds that have been fished commercially in the Clyde Sea area, Scotland. Sediment cores were taken to assess the population density of live maerl thalli prior to scallop dredging on marked test and control plots. These plots were then monitored biannually over a four-year period. Live maerl thalli were sparsely distributed at the impacted site, and experimental dredging had no discernible effect on their numbers. The previously unfished ground had dense populations of live maerl and scallops (both *Aequipecten opercularis* and *Pecten maximus*). While counts of live maerl remained high on the control plot, scallop dredging led to a >70% reduction with no sign of recovery over the subsequent four years. The vulnerability of maerl and associated benthos (e.g., the delicate bivalve, *Limaria hians*) is discussed in relation to towed demersal fishing practices.

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Key words: benthos, ecosystem effects, long-term impact, maerl beds, scallop dredging.

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Introduction

Widespread interest in the ecological effects of current fishing practices has stimulated intense research in recent years (Lindeboom and de Groot, 1998; Jennings and Kaiser, 1998). Recent studies have established that demersal fishing gear can have major immediate effects on the benthos (reviews by Dayton *et al.*, 1995; Auster *et al.*, 1996; Watling and Norse, 1998). This is particularly true of those types of gear that penetrate the substratum deeply, such as hydraulic dredges (Hall *et al.*, 1990; Pravanovi and Giovanardi, 1994) and scallop dredges (Thrush *et al.*, 1995; Currie and Parry, 1996; Collie *et al.*, 1997; Hall-Spencer *et al.*, 1999; Hall-Spencer and Moore, 2000).

Studies that involved monitoring the after-effects of gear impact on sedimentary habitats have often been unable to detect effects in the long term. The effects of gear impact on small areas within dynamic habitats, e.g. sandy sediments, are quickly diluted through

the immigration of benthos from the surrounding area or sediment redistribution (Hall *et al.*, 1990; Eleftheriou and Robertson, 1992). Even in large-scale studies, long-term effects of fishing activities have been difficult to differentiate from other sources of variability in the benthos, such as population heterogeneity, large seasonal fluctuations, and the effects of natural disturbances such as storms (Hall *et al.*, 1993; Currie and Parry, 1996; Kaiser *et al.*, 1998). In most cases, such studies have been undertaken in areas already modified by commercial fishing activity as there are now very few suitable grounds that have not been heavily influenced by towed fishing gear (Lindeboom, 1995; Tuck *et al.*, 1998). Consequently, benthic populations remaining in fished areas are generally resilient to gear impacts (Currie and Parry, 1996; Kaiser *et al.*, 1996).

Our study forms part of an investigation comparing natural variability with the effects of human activities on the ecology of maerl habitats in Europe (BIOMAERL

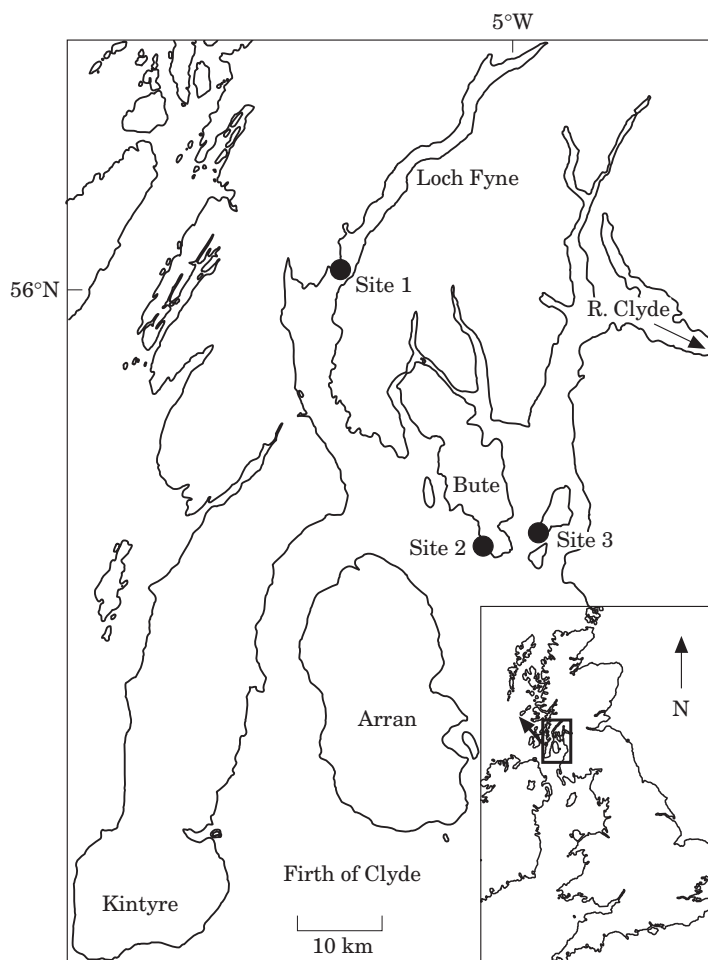


Figure 1. Locations of three study sites in the Firth of Clyde, SW Scotland, UK.

team, 1998, 1999; Grall and Glémarec, 1998; Hall-Spencer, 1998; Hall-Spencer and Moore, 2000). Maerl is a living sediment that typically harbours a high level of biodiversity (BIOMAERL team, 1998; Birkett *et al.*, 1998) but is very slow-growing in European waters (Potin *et al.*, 1990; Canals and Ballesteros, 1997). NE Atlantic maerl beds are usually characterized by coarse sediment, clean water, and significant bottom currents, and thus often provide good scallop fishing grounds. Concerns have been expressed about the sensitivity of this biotope to towed demersal fishing gear (MacDonald *et al.*, 1996; Hall-Spencer and Moore, 2000). Here, we report the difference between fished and previously unfished maerl beds on the west coast of Scotland and compare contemporary samples with those collected a century ago. Small-scale scallop dredging experiments are described, designed to determine: (a) the immediate physical impact on the habitat; and (b) the long-term effects on major habitat-structuring organisms in previously unfished and fished areas.

Methods

Three maerl grounds in the Firth of Clyde (Fig. 1) were surveyed in detail from 1994 to 1998 using a combination of Sprint[®] Remote Operated Vehicle (Perry Triton Ltd), RoxAnn[®], van Veen grab sampling (at least six grabs per site per quarter) and >200 h field observations using SCUBA. Site 1 was at Creag Gobhainn, Loch Fyne, where maerl occurred parallel to the shore along a gently sloping strip over an area of 17.5 ha from -6 m to -14 m CD (chart datum). This site had not been previously exploited by towed demersal fishing gear because of rocky outcrops, its proximity to shore, and the presence of a chartered telecommunications cable (laid in 1968) in a designated "trawling prohibited" area. Site 2 was situated on a shoal in Stravanan Bay ~0.5 km off the SW coast of the Isle of Bute where maerl covered an area of 6.75 ha from -6 m to -15 m CD. This site has been used by scallop fishermen over the past four decades, and scallop

dredging continued during 1994–1998. Site 3 was situated near the Tan Buoy, SW of Great Cumbrae Island, where maerl covered 4.0 ha from -6 to -10 m CD. The Tan Buoy area had also been dredged extensively over the past 40 years both by research vessels and by fishing boats. Full descriptions of these sites are given elsewhere (Hall-Spencer, 1998; BIOMAERL team, 1999).

Experimental fishing took place at Sites 1 and 2. First, surface and subsurface buoys were laid at -10 m CD at each site to provide permanent markers for the four-year study. Positions of these (determined by Magellan[®] Differential GPS) were $56^{\circ}00.601'N$ $005^{\circ}22.148'W$ (Site 1) and $55^{\circ}45.323'N$ $005^{\circ}04.265'W$ (Site 2). At each site, two buoys were laid 10 m apart to one side of the permanent buoy to delimit the width of an area to be fished (test plot) while the opposite side was untouched (control plot).

At each site in May 1994, eight replicate sediment cores (20 cm long, 10.3 cm in diameter) were taken by divers on the test plot, and a further eight were taken on the control plot. Collected cores were kept upright, frozen within 5 h of collection and stored at $-18^{\circ}C$.

The day after coring, test plots were fished from RV "Aora" (15 m, 260 hp) using a set of three Newhaven scallop dredges. The dredges were 77 cm wide, weighed 85 kg in air, and had 10 cm long, 0.8 cm wide teeth mounted 8 cm apart (nine per dredge) on spring-loaded tooth bars. A bag of linked 7 cm diameter steel rings extended behind each tooth bar to retain the catch. The dredges were towed once over each test plot for ~ 100 m giving fished areas of ~ 230 m².

Immediately after fishing, a pair of divers equipped with tape measures and writing slates recorded gross changes to surface topography on the test plots and took eight cores. Changes were also noted on the adjacent control plots where a further eight cores were taken. Labelled aluminium rods were pushed into the sediment to mark core positions. This coring routine was then repeated biannually over the next four years. The amount of living maerl on test and control plots was assessed as follows. Cores were first thawed and removed from the PVC pipes to note vertical stratification within the sediment. They were then wet sieved in fresh water through a 5 mm mesh. Maerl thalli that were alive at the time of collection (hereafter referred to as "live thalli") were removed with forceps and examined using a calibrated dissecting microscope. Live thalli had smooth surfaces and were either pink, owing to the presence of phycobilin pigments, or green if the phycobilins had leached out, revealing the presence of chlorophyll. Dead maerl fragments were sometimes smooth and white but more often had pitted surfaces, imbued with shades of brown and grey to black, depending on the degree of deposition of iron and manganese salts. A few dead fragments were pink or green owing to the

presence of boring algae (the Conchocelis stage of *Porphyra* spp. and *Ostreobium quekettii* Bornet et Flahault) but with pitted, eroded surfaces. Numbers of live thalli were counted in each core, and an estimate of the area they had covered was obtained by laying the plants in a monolayer on graph paper.

Maerl collected in 1885 from Site 3 was found with a collection of byssus nests of the bivalve mollusc *Limaria hians* (Gmelin) in the David Robertson zoological collection at the University of Glasgow. Three more maerl samples from Site 3 were found in the following boxed collections at the British Museum (London); Box 516 (code BM 000005147) labelled "*Lithothamnion calcareum* f. *compressa* Foslíe" collected by Mrs Robertson in 1891, Box 519 (code BM 000005180) labelled "*Lithothamnion squarulosum* Foslíe" and "*Lithothamnion corallioides* f. *australis*" collected by E. A. L. Batters in 1891, and Box 521 (code BM 000005182) labelled "*Lithothamnion squarulosum* f. *australis* Foslíe" collected by E. A. L. Batters in 1891. These archived samples were examined microscopically against the criteria outlined above to establish if the maerl had been alive at the time of collection. The thalli were measured (max. length) then identified using a JEOL-JSM 5200 scanning electron microscope, as described by Hall-Spencer (1994).

During 1995–1997, an extensive survey of Site 3 was made using hand-held cores, 60 van Veen grab samples, and making use of 11 h of diver observations to locate living maerl and *L. hians* to compare with collections from the last century. Live plants were washed in fresh water, air-dried, measured, and identified to species.

As a one-off investigation, a second scallop-dredging exercise was carried out at a distance from the test and control plots at Site 2 (-10 m CD; $55^{\circ}45.37'N$ $005^{\circ}04.22'W$). As before, two buoys were spaced 10 m apart to mark the north and south sides of a corridor to be fished. A series of circular plastic tubs (each 17 cm in diameter and 14 cm deep) were labelled and arranged in pairs by divers at approximately 0, 1, 2, 4, and 8 m on transects running at right angles to the towed corridor, i.e., north of the north buoy and south of the south buoy. They were left for 2 h to collect background levels of settling sediment, then sealed with water-tight lids and replaced with a second set of 20 tubs laid out in the same configuration. A gang of three Newhaven dredges was then towed from east to west along a ~ 100 m corridor between the buoys. After fishing, the proximity of the "0 m" sediment traps to the dredge track was measured and all tubs were sealed and retrieved 2 h after fishing. Sediment was allowed to settle, excess water was siphoned off, and the sediment was washed twice in distilled water. After resettlement, the supernatant was again siphoned off and the sediment dried in an oven at $50^{\circ}C$. Dried samples were then weighed and the largest

Table 1. Differences in seabed and benthos characteristics between Sites 1 and 2.

	Site 1 (Unfished)	Site 2 (Fished)
<i>Aequipecten opercularis</i> density (m^{-2})	0.27 ± 0.06	—
<i>Pecten maximus</i> density (m^{-2})	0.06 ± 0.02	0.01 ± 0.07
% <i>Pecten maximus</i> >7 years	83 (n=106)	0 (n=57)
Dredge tracks	None	Frequent
Byssus nests	Frequent	None
Molluscan species richness	107	82
Maerl cover (%)	25.3	1.8

Presence of commercial scallop-dredge tracks and byssus nests of *Limaria hians* as seen on diver and ROV surveys; number of mollusc species (from Hall-Spencer, 1998); live-maerl cover (m^{-2} ; in 30 0.1 m^2 grab samples); mean population densities (\pm s.d. in 85 and 224 1 m^2 quadrats for Sites 1 and 2, respectively) of two scallop species; and percentage of *Pecten maximus* over seven years old (n=number aged).

particle in each sample was measured using a dissecting microscope.

Results

Differences between sites

Although Sites 1 and 2 were similar in many respects (each maerl bed was located on a level area of seabed within sheltered parts of the northern Firth of Clyde with no differences in depth, salinity, temperature, or tidal amplitude), the seabed and benthos exhibited marked differences (Table 1) that reflected differences in exploitation history (Site 1: deployment of demersal fishing gear prohibited since 1968; Site 2: scallop fished for the past 40 years). Population densities of the scallops *Aequipecten opercularis* (L.) and *Pecten maximus* (L.) were higher at the unfished site than at the fished site, with *A. opercularis* absent from Site 2. Mature individuals of *P. maximus* predominated at Site 1, whereas no individuals older than seven years were observed at the fished site. Scallop-dredge marks were common on the fished site, and live *Limaria hians* were absent, although their dead shells were abundant. At the unfished site, *L. hians* had built roughly circular byssus nests (up to one metre in diameter) that stood 20 cm proud of the maerl bed surface.

Immediate and long-term effects of scallop dredging

Test plots at Sites 1 and 2 showed significant physical disturbance along a 2.54 m wide track with three parallel furrows corresponding to the width of each dredge. Natural bottom features (ripples, crab feeding pits, and megafaunal burrows) were eliminated along the tracks,

and boulders up to 1 m^3 had been dragged along the sediment surface. The granulometric structure of the surface sediment had shifted compared with adjacent, unfished areas. Mud and sand had been brought to the surface, and maerl gravel was sculpted into 3 cm high ridges at the edge of each dredge (Fig. 2). At Site 2, all of the *L. hians* nests in the path of the dredges were either torn and dragged along the seabed or were brought up in the dredge bags. Live maerl was buried up to 8 cm below the sediment surface, and biogenic carbonate structures (e.g., maerl thalli, echinoid test plates, and bivalve shells) were crushed and compacted. The control plots nearby were relatively unaffected, although some silt had settled on the upper surface of the maerl.

The sediment in cores taken prior to fishing was vertically stratified at both sites. Site 1 had a 1 to 2 cm thick "open lattice" layer of maerl gravel with a large amount of void space between particles. This open lattice overlaid fine sand and mud mixed with shell and maerl fragments. Site 2 had less fine material and was characterized by a 3 to 8 cm open lattice layer of maerl gravel overlying coarse sand with pockets of fine sand and mud mixed with shell and maerl fragments. Cores taken on control plots had the same pattern of vertical stratification throughout the study, but those taken on test plots immediately after fishing lacked vertical stratification. Thus test-plot cores lacked an open lattice layer and had less interstitial space and a greater proportion of fine particles at the surface. Sampling over the subsequent four years revealed a gradual return to a clean, gravelly upper layer of maerl at both sites, presumably due to winnowing away of fine material by water movement (during spring tides, bottom currents exceed 10 cm s^{-1} at both sites).

Figure 3 shows mean counts and area covered by live thalli in cores taken on test and control plots over four years. Live maerl was sparsely distributed in the previously fished area and experimental dredging had no discernible effect either on numbers or area covered. Live maerl was abundant in the previously unfished area and remained abundant on the control plot. There was no difference in the amount on the test plot immediately after scallop dredging as thalli that had been buried within the sediment were still alive. However, five months after dredging there were 70–80% fewer live thalli in the test cores than in cores taken prior to fishing. There were no signs of recovery in numbers or area covered over the following four years (Fig. 3).

The sculpted ridges and troughs of the dredge tracks remained visible within test plots for 2.5 years at Site 1 and 1.5 years at Site 2. They were gradually erased through bioturbation by large infauna (e.g., the thalassianid *Upogebia deltaura* (Leach) and the holothurians *Neopentadactyla mixta* (Ostergren) and *Thyonidium drummondii* (Thompson)) and feeding activities of gastropods, crabs, starfish, and fish. On shallow grounds



Figure 2. Appearance of a scallop-dredge track at Site 1 showing maerl scraped aside (centre of photograph) with muddy sediment brought to the seabed surface. A *Pecten maximus* scallop (arrow) has been dug out of its recessed position within the sediment.

at Site 2 (–6 to –8 m CD), commercial scallop-dredge tracks were erased by wave action during storms (Hall-Spencer and Atkinson, 1999).

Historical changes

Identification based on modern taxonomic criteria revealed that the variously labelled maerl thalli obtained between 1885 and 1891 at Site 3 were all *Phymatolithon calcareum* (Pallas) Adey et McKibbin. Microscopic examination showed that these plants had been alive at the time of their collection. *L. hians* nests collected in 1885 had >100 thalli up to 58 mm long entwined with byssus threads to form a protective mat. The three boxed collections at the British Museum had the following contents: Box 516, >100 thalli (up to 38 mm long); Box 519, 65 thalli (up to 50 mm long); and Box 521, 18 thalli (up to 43 mm long).

In 1995–1997, detailed surveys of the entire maerl bed at Site 3 yielded just 16 live *P. calcareum* thalli, all of which were smaller than those collected in the last century (<20 mm; Fig. 4). As with Site 2, dead shells of *L. hians* were common, but no live specimens were found.

Sediment trapping investigation

Only small amounts of sediment ($0\text{--}0.5\text{ g m}^{-2}$) were collected in traps deployed for 2 h before scallop fishing at Site 2 (Fig. 5), demonstrating the prevailing clarity of the water (12 m horizontal visibility recorded by divers). Immediately after fishing, however, suspended sediment had reduced visibility in the vicinity of the fished corridor to a few cm. Two hours later, suspended sediment had settled or dispersed with the current and the ~2.5 m wide scoured path along the maerl between the marker buoys was clearly visible to divers. The edge of the dredge track passed 1.5 m from the nearest sediment trap on the north transect and 6 m from the nearest trap on the south transect. Maerl around the path was blanketed by a newly settled layer of silt. Settled particles were largest in the tubs close to the dredge path (max. diam. 4.2 mm) and became progressively smaller with increasing distance (max. diam. 0.8 mm, 15 m away).

The relationship between the amount of sediment collected and distance from the track is shown in Figure 5. A minimum estimate of the amount of sediment transported to each side was derived by subtracting the area under the pre-dredge curve from that under the

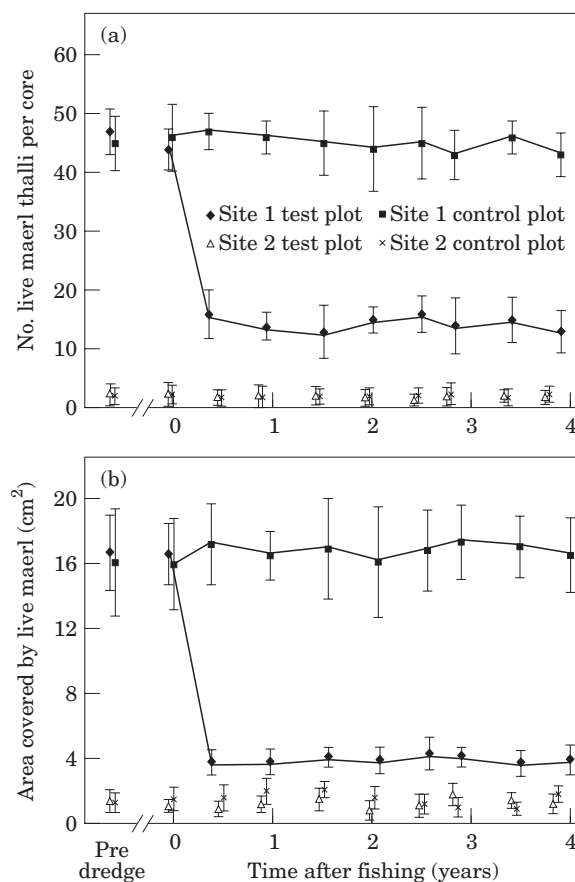


Figure 3. Live maerl thalli retained on a 5 mm sieve from 80 cm² sediment cores taken on test and control plots at Site 1 (previously unfished) and Site 2 (previously fished): (a) mean counts and (b) mean cover area (error bars are ± 1 s.d.; n=8).

post-dredge curve. This calculation indicates that at least 340 g of sediment was removed from each metre of the dredge track. The blanketing effect of the settled sediment was greatest close to the dredge path and was easily detected 15 m away where 4 g m⁻² of fine silt had settled.

Discussion

Studies on the fishing intensity and area swept by fishing gear in recent years (e.g., Kaiser *et al.*, 1996; Rijnsdorp *et al.*, 1998), indicate that most sedimentary benthic systems on the continental shelf of Europe have been modified by fishing activities in the last 100 years. In some places, such as the southern North Sea, fishing is now thought to be the main ecological structuring force on the benthos (Lindeboom and de Groot, 1998). As a consequence, it can be difficult to design robust field experiments to investigate the long-term effects of fishing

on unmodified communities because of the paucity of adequate control grounds (Lindeboom, 1995). Strategies that involve sampling close to seabed obstructions (Hall *et al.*, 1993), or in areas that are closed to fishing for other reasons (Tuck *et al.*, 1998; Hall-Spencer and Moore, 2000), do provide useful insights.

Had we restricted our investigation to previously fished maerl grounds (Sites 2 and 3), we would have concluded that although scallop dredges had significant effects in the short term (weeks), long term effects (over several years) could not be detected against a background of natural variation. However, observations on a previously unfished site have revealed significant long-term effects. This adds to an increasing body of evidence that the major impact of towed demersal fisheries occurs the first time an area is subjected to fishing pressure (Jennings and Kaiser, 1998).

Scallop dredging began in the Clyde Sea area in the 1930s with yields of <3 t yr⁻¹ (Elmhirst, 1945). Landings increased with the expansion of the fishery in the 1960s through the advent of more powerful boats, more efficient dredges, and better processing facilities (Mason and Fraser, 1986). Local catches per unit effort have since declined owing to dwindling stocks, although scallops remain the most important mollusc fishery in Scotland with 11 300 t landed in 1995 (Ministry of Agriculture, Fisheries and Food, 1996). A sustained impact over the past 30–40 years on local maerl beds is thus likely. Our historical data from the Tan Buoy show extensive changes over the last 100 years, from a living maerl bed with abundant large thalli and nests of the gaping file shell *L. hians* to a bed of predominantly dead maerl with few, small, live thalli and no *L. hians*.

Our observations on the immediate physical impact of scallop dredging on maerl grounds are consistent with those observed on other types of sedimentary habitat (Caddy, 1973; Eleftheriou and Robertson, 1992). Our calculations of sediment erosion (340 g m⁻¹ of dredge track) and of the area blanketed by redistributed sediment (12 times the area that had experienced contact with the gear) are underestimates, because sediment transported more than 15 m away was not quantified. Newly settled sediment, however, appeared to have no lasting effect on the benthos adjacent to the dredge tracks. The fauna are likely to be adapted to the periodic sediment redistribution that occurs in shallow sublittoral habitats (Hall, 1994; Hall-Spencer and Atkinson, 1999).

The differences between unfished and exploited areas confirm that maerl grounds are especially fragile habitats. European legislation under Council Directive 92/43/EEC on the Conservation of Natural Habitats and of Wild Fauna and Flora (1992) specifies that exploitation of maerl habitats must be compatible with their maintainance at a "favourable conservation status". This may not be the *status quo* for many European maerl beds because most Clyde grounds,

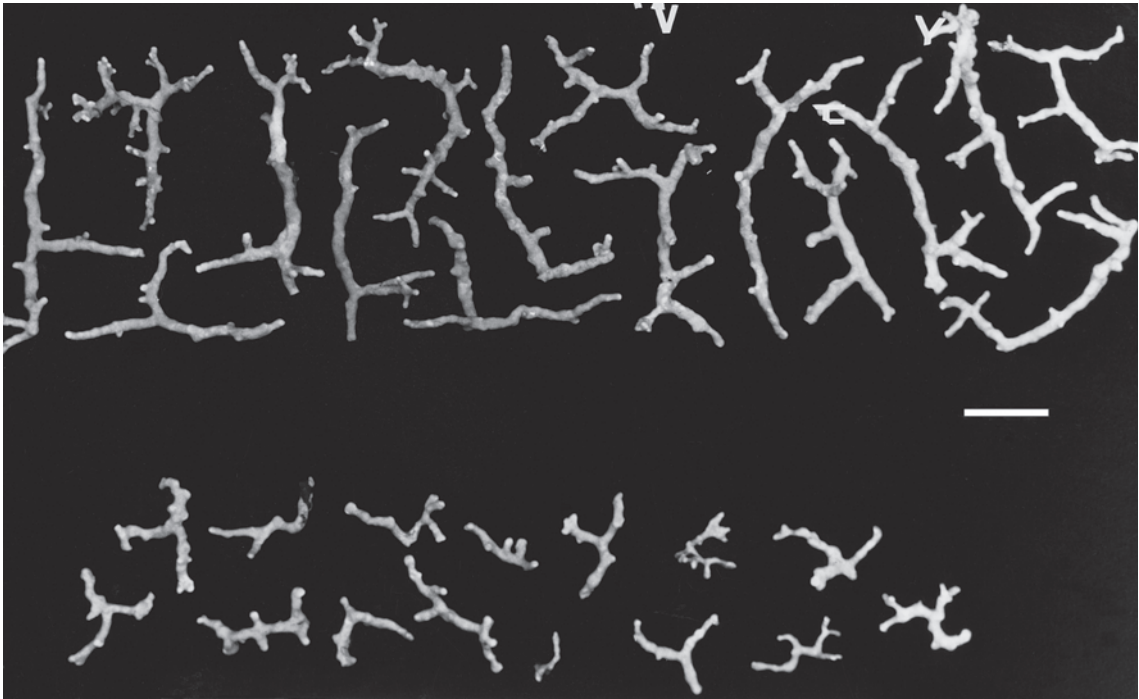


Figure 4. *Phytomatolithon calcareum* maerl thalli collected at Site 3 by Batters in 1891, before scallop fishing began in the Clyde Sea area (upper set), compared with specimens collected from the same site in 1995 (lower set). Scale bar represents 1 cm.

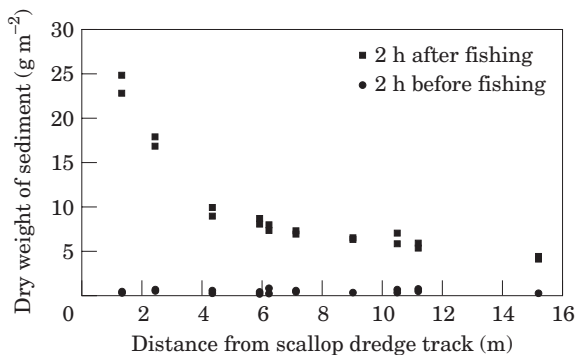


Figure 5. Sediment collected in traps 2 h before and 2 h after scallop dredging at different distances from the track (Site 2).

at least, have been extensively modified by scallop dredging.

Maerl beds belong to the most sensitive habitat type compared with other sedimentary grounds, shifting sands especially, where evidence of towed gear impacts is ephemeral (Hall, 1994; Curry and Parry, 1996; Kaiser *et al.*, 1998; Jennings and Kaiser, 1998). Parallels may be drawn with the effects of towed demersal fishing gear on *Modiolus modiolus* (L.) communities in the Irish Sea (Magorrian *et al.*, 1995) and seagrass communities in the Mediterranean (Martin *et al.*, 1997), or with blast fishing

on coral reefs (Jennings and Polunin, 1996), where impacts are long lasting because they affect the survival of key habitat-structuring species with poor regenerative abilities.

The integrity of maerl habitats depends upon the survival of a surface layer of slow-growing algae. These algae are unable to withstand prolonged burial owing to lack of light and so they are easily killed by scallop dredging. It is thought provoking that our single tow of three dredges ($\sim 230 \text{ m}^2$ ground contact) had effects (sediment redistribution, live maerl burial) that were clearly discernible four years after the event, because the experimental design of our impact study was deliberately limited in scope. The area disturbed by a commercial scallop boat that typically tows 16 dredges has been estimated to be 6.6 km^2 per 100 h fishing (Kaiser *et al.*, 1996). Clearly, repeated scallop dredging on the ground studied would considerably reduce the amount of living maerl (given thallus burial, comminution, and smothering by silt). However, caution is needed before extrapolating our results to other maerl-bed areas and gear types. Preliminary work on Maltese maerl beds (BIOMAERL team, 1999) has indicated that commercial otter trawling has had no negative impact on the cover of live thalli. The relative hardness of the Maltese ground coupled with a lack of silt and robust, rounded maerl thalli appear to confer resistance to periodic otter trawling.

The lack of discernible recovery of a previously unfished maerl bed in Scotland over four years is related to the slow growth (Potin *et al.*, 1990; Canals and Ballesteros, 1997) and poor recruitment of maerl species. *Phymatolithon calcareum*, which is the main species in the Clyde Sea area and has the widest distribution of the European maerl-forming species (BIOMAERL team, 1999), rarely produces reproductive spores, and newly settled thalli have never been found in the British Isles (Irvine and Chamberlain, 1994). Our findings add impetus to moves to identify pristine maerl beds for conservation in Europe. In SE Spain, the Tabarca Island marine reserve has worked well in protecting local maerl beds from the effects of towed fishing gear (Ramos, 1985; BIOMAERL team, 1999), and a project is currently underway to establish management schemes for four candidate "marine Special Areas of Conservation" containing maerl beds within the UK (Birkett *et al.*, 1998).

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